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Assessing the Role of Economic Instruments in a Policy Mix for Biodiversity Conservation and Ecosystem Services Provision: A Review of Some Methodological Challenges

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Abstract: In this paper we review a number of methodological challenges of evaluating and designing economic instruments aimed at biodiversity conservation and ecosystem services provision in an existing policy mix. In the context of the EU 2010 goal of halting biodiversity loss, researchers have been called upon to evaluate the role of economic instruments for cost-effective decision-making, as well as non-market methods to assess their benefits. We argue that cost-effectiveness analysis (CEA) and non-market valuation (NMV) methods are necessary, but not sufficient, approaches to assessing the role of economic instruments in a policy mix. We review the principles of "social-ecological-systems" (SES) [1] and discuss how SES can complement economic cost and benefit assessment methods, in particular in policy design research. To illustrate our conceptual comparison of assessment methodologies, we look at two examples of economic instruments at different government levels – payments for ecosystem services (PES) at farm level and ecological fiscal transfers to municipal /county government. Which policy design issues can cost-effectiveness analysis and non-market valuation methods address for these two instruments? What conceptual problems are introduced when evaluating policies in an instrument mix? How can the SES framework complement CEA and NMV in policy assessment and design? We draw on experiences from Brazil and Costa Rica to exemplify these questions. We conclude with some research propositions.

Keywords: biodiversity, ecosystem services, policy mix, social ecological systems, payments for environmental services, ecological fiscal transfers

1. Current Economic Policy Assessment Criteria and Their Limitations in a Policy Mix

Real-life policymaking is seldom a neutral search for the single most optimal instrument to maximise welfare [2]. In no other environmental policy-making field is this truer than biodiversity conservation and ecosystem services provisioning. Despite this recognition, recommendations by economists for policy mix design in the environmental field have mostly been written based on the nature of pollution issues, focusing on a limited set of policy design principles that have been amenable to economic modelling [3, 4]; achieving maximum marginal benefits (environmental effectiveness criterion), minimisation of marginal costs of achieving a given policy objective (cost-effectiveness criterion), equating marginal benefit with marginal costs (cost-benefit or efficiency criterion).

Comparisons of the effectiveness of a particular category of conservation policy or project has a number of data and methodological challenges (see e.g. [5-7], as does accounting of any given conservation policy's opportunity, implementation and transaction costs [8-10]. Valuation of the benefits of biodiversity conservation has been challenged both on grounds of theoretical consistency (e.g. Spash, 2008), and methodological feasibility (e.g. [11]. Cost-effectiveness (CEA) or cost-benefit (CBA) analysis ranking of alternative "instruments" is therefore already a formidable task, assuming that internal methodological limitations and data gaps can be overcome at a particular location.

To these optimization criteria, economists often emphasise other sub-goals that support welfare maximization, particularly incentive compatibility, dynamic effects and innovation, distributional and equity concerns, revenue generation, compliance cost control or administrative feasibility [12] [3] [2]. The task of evaluating policy "instruments" then moves conceptually from optimization on the margin, or ranking along a single Cost-Effectiveness or Benefit-Cost ratio, to ranking alternatives using multiple criteria assessment (MCA) techniques [13].

A conceptual problem with CEA, CBA and even MCA approaches for evaluating alternative economic instruments arises in their use in evaluating policy mixes; policy combinations may be difficult to define as independent alternatives. Multiple impact assessment criteria – such as policy effectiveness, cost, distributional equity, legitimacy – are likely to be correlated across proposed economic instruments when an existing policy mix constitutes a common governance context within which the alternatives are being evaluated. For example, transaction costs may be broadly similar across alternative PES schemes, because they will be implemented with the same regulatory framework of an established forestry law.

Reviews of economic instruments in conservation hint at these problems. A recent review of biodiversity policy instruments has found little empirical evidence or reporting of the impacts of alternative policy instruments for forest biodiversity conservation [14]. According to the review, environmental outcomes are rarely assessed, and if so not in relation to costs of implementing a policy relative to a baseline. Clear recommendations of when and where to use 'market-based instruments' instead of, or complementarily to, more hierarchical command-and-control approaches are hard to formulate because biodiversity is such a heterogeneous good - policies need to be tailored to local site specific needs [15]. Both reviews point to the lack of consistency in evaluating cost-effectiveness of policy instruments, suggesting methodological weaknesses in previous studies, or prohibitive data collection costs given the difficulties in transferring existing results across heterogeneous sites.

2. Studies of Cost-Effectiveness and Efficiency of Economic Instruments

A handful of papers suggest that the lacking conclusions regarding market-based instruments may be more than a 'measurement problem'. Analysis of policies for biodiversity conservation needs to deal simultaneously with several sources of market failure [7] and therefore the implementation of various instruments needs to be considered simultaneously. Economic instruments are usually introduced and applied in contexts in which various command-and-control regulations pre-exist [16]. Complementarities between instruments have been noted, for example between payments for environmental services (PES) and regulation [7, 17]. In fact, most PES schemes work with a mix of other policy instruments (Box 1). Other examples are voluntary conservation schemes which have been found to be effective when there is a threat of regulation in the background [18], and a number of complementarities in combining the "stick" of regulation and the "carrot" of market based instruments have been pointed out [19]. Further, as economically optimal policies can often not be defined, a "second-best" solution is searched on the combination of instruments [15, 20]. These issues complicate cost-effectiveness and efficiency assessments as one-on-one comparisons or rankings of policy instruments in a particular setting are severely limited.

Box 1. What is a policy mix?

Vatn (2005) distinguishes between three main categories of policy measures: *Economic instruments*, based on the assumption that agents, be they individuals or institutions, are rationally calculative; *Legal instruments*, assuming both rationally calculative and social or normative reasoning; *Informational instruments*, operating through cognitive and normative processes, and an assumption of agents' bounded rationality. In the context of designing payments for environmental services, Vatn (2008) frames the question of policy design as fundamentally a search for combination of alternative governance structures (hierarchies, markets, and community management) that can be used to handle ecosystem interconnections that cause externalities across socially defined property and use rights boundaries. Jordan et al. (2010) uses the concept of *policy repertoires*. Turner (2005) describes a *policyscape* as "policies and personal preferences influencing the spatial productivity, richness, and uses [of the landscape]". Bressers and O'Toole (1998) argue that *policy actors' network* influence the feasibility of an instrument through interconnectedeness and policy learning - the more an instrument help maintaining the existing features of the network the more likely it is selected during the policy formation process.

The relevant geographical areas for evaluating the mix of instruments can be defined by the different governance levels (governmental levels, market, community). We will take a policy mix to mean combinations of multiple types of 'instruments' in a particular geographical area. We will argue that for purposes of empirical analysis and policy design, a policy mix should be decomposed into a *portfolio* of institutional characteristics. A research proposition is that cost-effective policy mix is one where the spatial configuration of its portfolio of characteristics is correlated to biodiversity and ecosystem service characteristics across the landscape. Furthermore, policy actors networks are more interconnected with proximity. A policy mix is more effective where administrative boundaries match spatial configuration of ecosystem services, because of policy actors proximity and greater congruence with the actor network.

Recent theoretical and case study review work, particularly on PES, has identified a number of feasibility criteria for the success of economic instruments. For example, Ferraro and Simpson (2002) use a model to identify the conditions under which direct conservation payments are more cost-effective than indirect subsidies. These conditions include available data to target land-uses providing ecosystem services (ES), low transaction costs, incentive compatible contracts, and enforced property rights. Several of the assumptions of the model are themselves legal and informational instruments, i.e. institutions for environmental monitoring, enforcement, contract law and property rights. From a conceptual point of view there is only a limited sub-set of characteristics of different economic instruments which are feasible alternatives, since all economic instruments build on some form of institutional context or pre-existing regulation.

Based on a detailed review of PES schemes, Engel et al. (2008) identify some generic instrument assessment criteria [21], which additionally highlight the complexity of factors that have to be considered when evaluating economic instruments for conservation in their policy context. Wunder discusses where and when conservation payments work on the margin [19][22], and which are necessary economic, competitive, cultural, institutional and informational preconditions for PES [17]. He points out that the introduction of market incentives into a setting where PES conditions on the margin are not met can lead to PES undermining protected areas that are working relatively effectively. Further limitations to the implementation of economic instruments, such as PES, can be site specific such as communities' requirements for trust-building, or the unacceptability of *quid pro quo* agreements in donor crowded areas where conservation funding is easily available [19].

While these are examples of theoretical or *ex post* assessments, authors call for more work on *ex ante* assessment, baseline definition and incorporation of monitoring in the policy design. Before detailed design of PES is undertaken, assessments of the efficiency of existing policy instruments and of pre-existing motivations for environmental service provision should be undertaken [19].

What then of non-market valuation in support of policy mix assessment in the field of biodiversity conservation? Stated preference methods use surrogate indicators to reduce the complexity of biodiversity and ecosystem services, evaluating how these are perceived and valued by landusers (e.g. [11, 23]. Although limited in the complexity they can convey, valuation methods are regularly used to evaluate perceptions of and quantify willingness-to-pay for increased conservation. In principle, economic valuation of the benefits of biodiversity conservation has several roles to play, although we argue that these roles are limited in practice:

Raising awareness: Brown and Moran (1994) call for valuation of biodiversity to raise "common concern". Using benefit transfer, global level cost-of-policy-inaction studies have been made possible [24]. Non-market valuation results, transferred and aggregated at a global scale have played an important role in raising the policy visibility of biodiversity conservation relative to other social issues.

Regulatory cost-benefit analysis: We have argued above that evaluating economic instruments in a policy mix poses a problem of defining meaningful policy alternatives. The answer to the question "is PES a cost-effective biodiversity conservation instrument?" in case study reviews is often that it depends on the ecosystem service in question and the social and institutional setting [7]. Assessment and design of economic instruments for biodiversity conservation aims to respond to heterogeneity in the provision of services at landscape and farm level scales, so cost-benefit analysis of any particular economic instrument loses most of its meaning for decision-support at aggregated national level.

Spatial targeting: determining which land uses provide the biggest incremental biodiversity benefit compared to land-owners opportunity costs. Not being able to link changes in biodiversity and ecosystem services to changes in land use is identified by some economists as one of the principle limitations to environmental valuation being used as a conservation policy assessment tool [14]. To other critics, indicators that capture the spatial heterogeneity of biodiversity and ecosystem services cannot be expressed in a meaningful way to the public, and so cannot be subject to meaningful statements of preference or valuation [25]. Wunder (2007) points out that the costs of evaluating baselines and effectiveness of policies on ecosystem services such as watershed services may easily exceed the aggregate benefits of the service itself in small watersheds. He advises rough farm-level opportunity cost calculations will often suffice to evaluate the feasibility of implementing PES practices. Non-market valuation at the spatial scale useful for targeting different economic instruments adds another layer of information costs to the often substantial costs of biophysical baseline and impact analysis. Benefits transfer seems initially to be a low-cost solution to this problem. However, meta-analyses of biodiversity valuation studies are dominated by variables of study design, with variables describing biodiversity impact, local population and institutional characteristics in minority (e.g. [26]).

Non-market valuation and benefit transfer has a very significant research challenge in addressing the concerns of spatial heterogeneity of biodiversity conservation. While this challenge is being met, operational conservation policy needs to find a balance now in using surrogate biophysical indicators to assess biodiversity conservation effectiveness that are at the same time complex enough to be sensitive to land-use change, and simple enough to be meaningful to land-users and regulators. Advances in geographic information systems and remote sensing however, are rapidly reducing the cost of such evaluations and are proving useful in providing the link between the ecological component of ecosystem services, and the economic component, particularly in identifying which portions of the landscape are most capable of providing specific services (or where the service is most greatly needed [27]. We turn to this issue in the next section, before discussing a general framework for policy mix assessment in social-ecological systems.

3. The Measurement of Biodiversity and Ecosystem Services for Policy Assessment

Any assessment of the marginal benefits of conservation measures requires that conservation objectives are operationalised through a set of indicators that are appropriate to quantify gains. The science of systematic conservation planning (SCP) has developed a body of theory and methods to quantify how a set of areas on which conservation policies are implemented represent natural values, as well as which is the marginal contribution to biodiversity representation of each individual area (e.g. [28]). The key metrics of marginal gain is *complementarity,* i.e. the marginal contribution of an area to biodiversity representation [29]. The calculus of complementarity is based on various quantifiable indicators, typically features of the landscape that correspond with the patterns of distribution of natural systems [30].

The potential to estimate marginal gains of conservation efforts opens opportunities to link policy effectiveness analysis with the body of theory and methods developed by SCP. However, a series of challenges remain to be solved before the methods can be effectively integrated into policy design and evaluation. The first constraint is that conservation objectives often go beyond that of biodiversity representation at national or regional scales. They are generally stated at various governance levels, spanning from those comprised in international agreements (i.e. Convention on Biological Diversity, Ramsar Convention), through national laws and directives, down to the management plan of a particular unit of land dedicated to conservation activities. Further, conservation values assigned to a particular area are typically multiple and of various kinds such as area size, degree of naturalness, provision of habitats for endangered species, representation of particular bio-geographic regions and protection of water courses.

More recently, criteria other than representation, and associated with the viability and probability of persistence of biodiversity attributes in the landscape [31, 32], have been incorporated into SCP approaches. These advances widen the scope from biodiversity representation to that of the maintenance of fundamental ecological processes (e.g. dispersal, colonization, meta-population dynamics) associated with the long term persistence of biodiversity [29]. Surrogate biodiversity indicators often target the provision of particular services or benefits [33]. The methodological integration of these criteria within surrogate indicators of biodiversity complementarity/representativeness is incipient [34], for example in research of on-farm biodiversity conservation in agroforestry [35-37]. These methodological advances based on spatial multiple criteria analysis, can constitute new opportunities to simultaneously evaluate the outcome on multiple biodiversity indicators without conducting non-market valuation or cost- benefit analysis.

Although there is an increasing amount of evidence establishing a correspondence between functional diversity and ecosystem service provision [38] [39, 40], there may be trade-offs between ecosystem service provision and other biodiversity conservation priorities (e.g. conservation of endangered species) [33]. MCA is appropriate for the consideration of different choices or when there exist a number of criteria which conflict to a substantial extent[41]. The approach can therefore provide an analytical framework that enables a combined assessment of the outcome of policy instruments considering objectives of biodiversity conservation, ecosystem service provision and forms of sustainable use of nature which may conflict with each other.

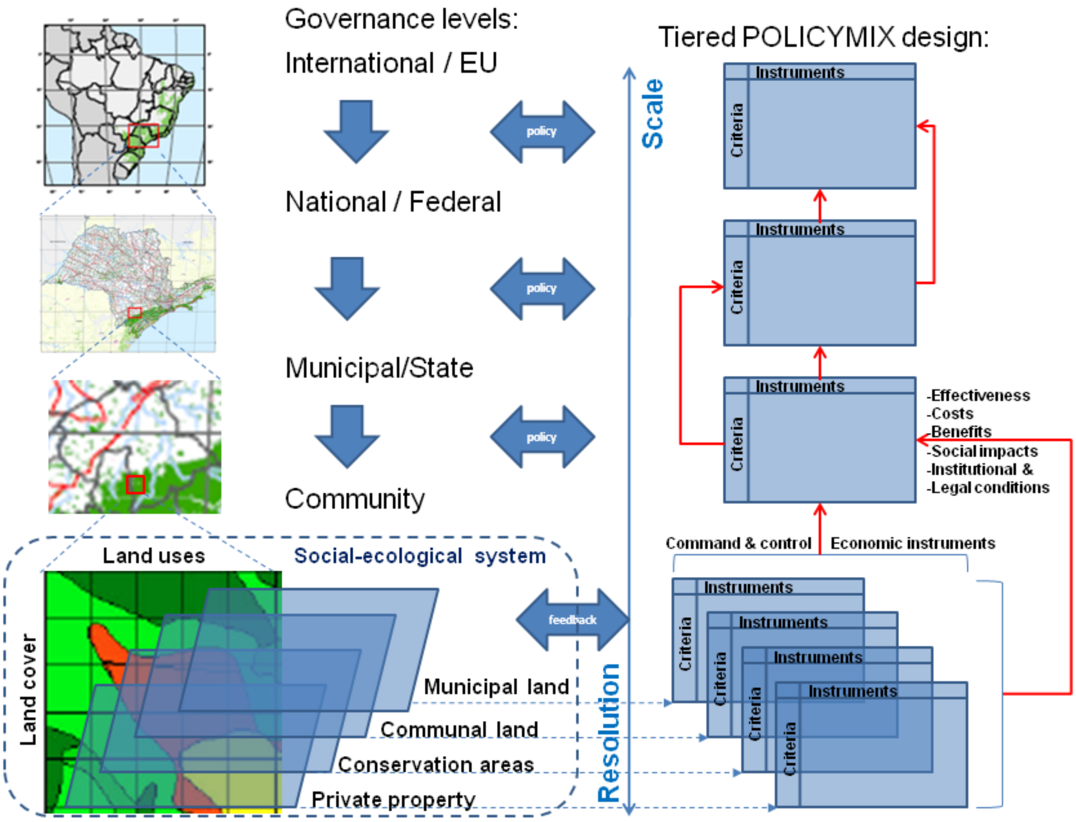
The most challenging constraint to the assessment of the effectiveness of a mix of conservation instruments is the kind and quality of the data available to assess change across multiple spatial scales and conservation objectives. Surrogate biodiversity indicators are often derived from various types of digitised cartography (climate, vegetation, bedrock, land-form maps), available at rather large scales [27, 42]. This restricts the analysis to large land units at regional scales. Recently, remote sensing data and geo-referenced data bases have considerably improved data availability to describe and quantify natural variation [29]. However, economic instruments aimed at farm property/forest stand level are implemented at small-spatial scales, which requires high resolution/high accuracy data to ensure robustness and transparency in the spatial targeting, monitoring and evaluation.

4. A conceptual Approach to Policy Mix Analysis: Social-Ecological-Systems Framework

There can be a disjunct in the scale of impact of policies adopted at different levels and those needed for the conservation of a particular region of ecological interest [43]. This disjunct may generate unintended spatial externalities. Effective biodiversity governance therefore has to consider the spatial characteristics of conservation benefits and costs in relation to governmental levels [44, 45]. Figure 1 illustrates the challenge of carrying out an impact assessment regarding any particular instrument in the context of multiple levels of government and ecological scales.

In rational planning, policy-making at higher spatial scales of governance determines which instruments may be legitimately implemented at lower levels of administration. In the local level context the social-ecological system determines the outcomes of the instruments. However, policies operate as a multi-level governance process, which makes policy implementation more context-relevant, but also creates discontinuities and problems of scale mismatch [45-47]. For example, performance assessment criteria forseen by the instrument mat be at a higher spatial resolution than is possible based on data monitoring systems [48]. Resulting uncertainty in policy implementation is best dealt with using experimental or adaptive policy design [2, 49], where impacts at lower tiers iteratively inform design at higher tiers [1].

Figure 1. A multi-tiered policy impact assessment framework needs to address interaction across instruments due to common governance structures of apparently alternative instrument; be robust to correlation across assessment criteria due to spatial interactions (externalities) between land-uses that are subject to policy mixes. Case-based, rather than theory-driven assessment assumes that experiences on the ground determine policy design at higher governance levels.



Research on the role of economic instruments in a policy mix therefore requires a conceptual framework that can work at several scales, across cases and be adaptable to the complexity of social and ecological context in each case. It also needs empirical tools to implement such a framework that strike a balance between case specificity, generalisation and transferability. Research on governance in social-ecological systems advocates a tiered approach to evaluation of broad system variables, focusing empirical effort on higher resolution variables only where necessary[1]. It suggests that social-ecological systems variables can provide the basis for evaluating the transferability of policy assessment results across geographically different case study contexts.

Ostrom's (2007) SES framework identifies first-tier variables as Resource System (RS), Resources Units(RU), Governance System (GS) and Users (U) and Interactions (I) and Outcomes (O) between these. Table 1 provides an overview over first and second tier variables in Ostrom's SES framework. The variables are listed without reference to any particular type of resource system.

Table 1. A comparison of variables across different empirical approaches to the analysis of forest conservation policies.

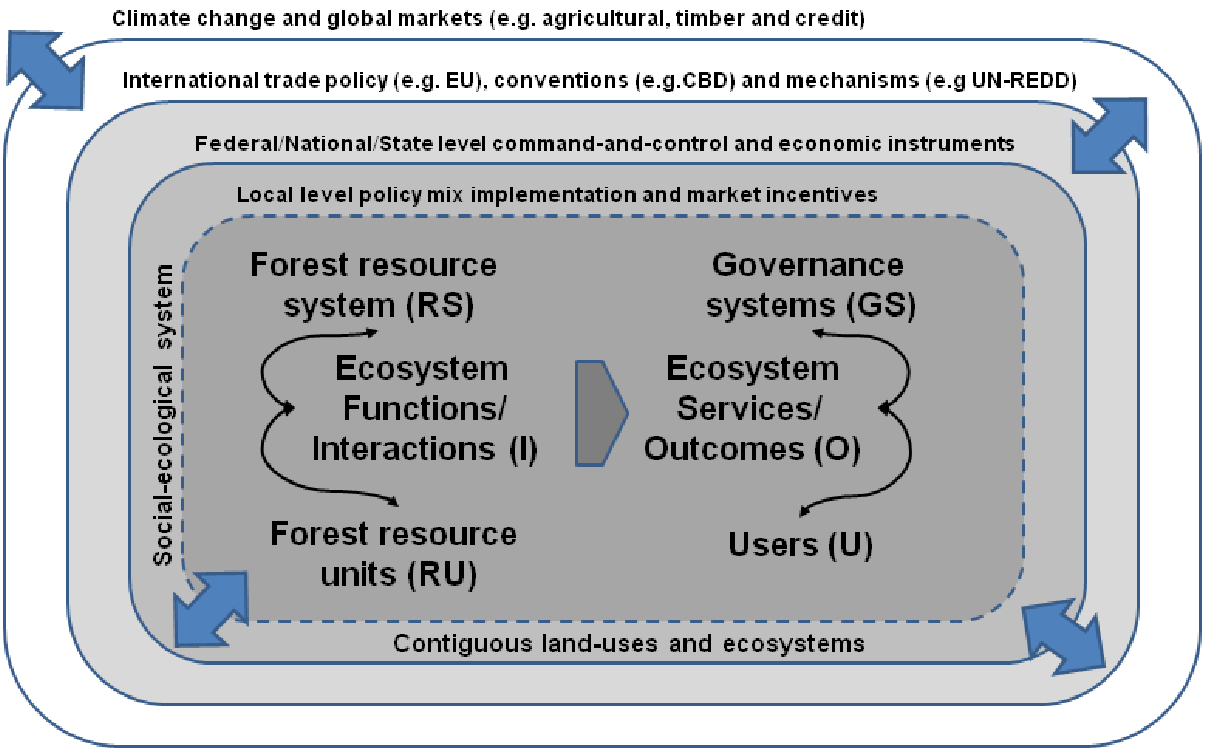
Ostrom(2007)		Brooks et al. 2006	Padgee et al.2006	Wunder, Engel, Pagiola (2008)	Angelsen (2007)
	Social-ecological systems (SES)	Integrated Conservation and Development Projects (ICDPs)	Community Forestry Management (CFM)	Payments for Environmental Services (PES)	Spatial land rent models
Resource System (RS)	RS1- Sector (e.g., forests) RS2- Clarity of system boundaries RS3- Size of resource system RS4- Human-constructed facilities RS5- Productivity of system RS6- Equilibrium properties RS7- Predictability of system dynamics RS8- Storage characteristics			Spatial scale and current size	Yield (y^i) = f^i (l^i,k^i, agroecological conditions: soil quality, rainfall, temperature) Capital k^i = k(technologies applied) v = v(extent and quality of roads, rivers, general infrastructure)
Resource Units (RU)	RU1- Resource unit mobility RU2- Growth or replacement rate RU3- Interaction among resource units RU4- Economic value RU5- Size RU6- Distinctive markings RU7- Spatial & temporal distribution				Yield (y^i) = f^i (agroecological conditions: soil quality, rainfall, temperature
Users (U)	U1- Number of users U2- Socioeconomic attributes of users U3- History of use U4- Location U5- Leadership/entrepreneurship U6- Norms/social capital U7- Knowledge of SES/mental models U8- Dependence on resource	Market integration - market sales - market purchase - wage labor - threat (type of use and severity) Homogeneity of population	Skills and experiences Strong leadership and effective organizing (present/absent)	Who else benefits?	Labour l^i = l(technologies applied) p^i = p(distance to centre, market demand) w = w(distance to centre, off-farm employment opportunities, migration) q= q(distance to centre) c^i = c(distance to centre) v = v(distance to centre)
Governance System (GS)	GS1- Government organizations GS2- Non-government organizations GS3- Network structure GS4- Property-rights systems	Utilization/protection - IUCN ranking - use permitted Decentralization	Tenure security Clear ownership Clearly defined boundaries Designated areas	Environmental services targeted Environmental services paid for Who buys? Who sells?	c^i = c(property regime, law enforcement/corruption, land competition)

Governance System (GS) (cont.)	GS5- Operational rules GS6- Collective-choice rules GS7- Constitutional rules GS8- Monitoring & sanctioning processes	-implementation level - local decision-making - targeted beneficiaries	Congruence biophysical and social boundaries Rules and regulations Effective enforcement Monitoring Sanctions Local responsibility Local autority (present/absent)	Who initiated? Intermediaries External donor support Seller selection Monitoring Sanctions Conditionality Payment (mode, amount, timing Differentiation) Contract duration	
Interactions (I)	I1- Harvesting levels of diverse users I2- Information sharing among users I3- Deliberation processes I4- Conflicts among users I5- Investment activities I6- Lobbying activities			Obstacles to implementation	Market prices (p^i) Wage rates (w) Annual costs of capital (q) Cost of enforcing property rights (c) Access costs (v)
Outcomes (O)	O1- Social performance measures (e.g., efficiency, equity, accountability) O2- Ecological performance measures (e.g., overharvested, resilience, diversity) O3- Externalities to other SESs	Ecological Economic Attitudinal Behavioral (failure, limited success, success)	Success/failure		Land rent: $r^i (d) = p^i(d) f^i(l^i, k^i) – w(d)l^i – q(d)k^i – c^i (d) – v(d)$
Social, Economic, and Political Settings (S)	S1- Economic development. S2- Demographic trends. S3- Political stability. S4- Government settlement policies. S5- Market incentives. S6- Media organization.			Linked to other policy tools?	p^i = p(taxes, subsidies, export/import regulations, marketing efficiency(competition)) w = w(economic development) q= q(access to credit, interest rates)
	Social-ecological systems (SES)	Integrated Conservation and Development Projects (ICDPs)	Community Forestry Management (CFM)	Payments for Environmental Services (PES)	Spatial land rent models (i= index for land use sector)

Note: for each of the studies we have tried to organise explanatory variables based on the greatest similarities with SES framework second tier variables. The exercise is meant to generate discussion on the complementarities between policy analysis approaches, as well as on the causality /hierarchy of explanatory variables for forest cover change, and in turn biodiversity conservation and ecosystem services provision.

In Figure 2 we put the SES framework in the context of evaluating a policy mix at different levels for management of forests and adjoining land-uses[50]. The SES framework implies no causality. The SES framework is a menu of variables for a diagnostic of which resource regimes are sustainable and which are not. The SES framework makes no causality assumptions in the interest of avoiding 'panacea' policy solutions and is an advantage in putting different disciplines on an even footing[1]. Causality becomes a concern once interest turns from ex post diagnostic to prospective impact assessment[51].

Figure 2. A multi-tier and multi-scale framework for analyzing the impacts of economic instruments in policy mixes on social-ecological systems.



In Figure 2 we make a distinction between ecosystem functions as interactions in the biophysical system and ecosystem services as outcomes. Which ecosystem functions are called ecosystem services is the result of use and governance systems. This is an interpretation of the SES framework which implies that ecosystem services are the outcomes or social objectives of interest to policy analysis [52] [33][53].

SES suggests a framework for systematic comparisons of policy context. Table 1 illustrates that while second tier variables provide additional detail, lower-tier variables and measurable indicators are needed if SES framework is to be used for cross-case comparisons and describing sustainability characteristics of policy mixes. As an example, Table 1 compares SES second-tier variables with the criteria used by Brooks et al. (2006) in their study of 28 Integrated Conservation and Development Projects (ICDPs); the Pagdee et al. (2006) study of 69 Community Forest Management (CFM) cases; the Wunder et al. (2008) study of 14 PES cases; and the Angelsen (2007) model of land rent. In table 1 we sort the significant variables found in these studies under the respective headings of the first-tier variables in the SES framework. Similar comparisons with other quantitative studies on conservation policy success/failure would point out what parts of the SES framework are most easily subject to systematic cross-case assessment. The significance of certain site characteristics for project or management success found by Brooks et al. (2006) and Padgee et al. (2006) suggest that the task is empirically feasible in practice despite case heterogeneity.

The SES framework falls short of being a methodology for comparing the role of any particular economic instrument in, or the relative sustainability of, alternative policy mixes. An approach to decomposition of policy criteria and conditions in environmental policy design is needed. This is not novel, but comparisons of ‘economic instruments’ against one another have sometimes blurred the distinction between contextual conditions required for a policy to work, and objectives of the policy [2]. Also, given the challenges in defining economic instruments used in conservation, some form of decomposition of policy characteristics seems necessary for the purposes of evaluating a policy mix[54]. Brooks et al. (2006) recognised that although specific projects become known for specific strategies, in practice more than one strategy is used in any one project. Accordingly they focused not on the strategies or policies per se, but on the assumptions underlying the strategies currently embraced by conservation organisations, further detailing sub-goals for success/failure. Despite the differences in conceptual models of numerous studies evaluating forest management success/failure, Padgee et al. (2006) successfully used the SES framework to describe diverse policy settings in some detail. There may therefore be advantages to combining the SES diagnostic approach with environmental policy design criteria of the environmental economics literature[55].

A fleshing out of the governance system characteristics (see Table 1) of SES with observable “design principles”[56] of the self-organising common-property resource management (CPRM) institutions [57], and criteria for environmental policy selection from environmental economics literature [3, 12, 58] [2] may be a basis for a generic hierarchy of policy ‘traits’ with observable indicators. A comparison of the institutional design criteria listed in the footnotes immediately show criteria operating at different levels of government/governance and spatial scales. The analysis of the performance of economic instruments therefore needs, at the very least a hierarchy of policy evaluation criteria. In order to account for differential importance of these criteria at different spatial/governance levels, criteria relating to implementation at the local level would need to be nested within national policy criteria. Ostrom (2007) refers to “nested conceptual maps”. At any governance level institutional “traits” can also be further decomposed into elements of an institutional “grammar” [59, 60]. There is a need to adopt a common nomenclature for describing the “traits and functional characteristics” of legal, informational and economic instruments. Such a nomenclature would make it possible to describe a portfolio of successful “traits” and “functions”, rather than combinations of instruments per se. The principle of SES diagnostic is to use variable levels of detail depending on policy-design needs and the information available.

Ostrom (2007) advocates nested conceptual maps to deal with the heterogeneity of “situation variables”[61] affecting the sustainability of governance systems. While the SES framework is a rich conceptual approach, hypothesis testing requires some form of quantitative analysis that can deal with variable data types and sub-sample sizes at different hierarchical levels[62]. The multi-dimensional nature of conservation policy success also complicates model specification, with hypothesis testing facing multi-dependent variables [6]. In the studies we reviewed, the variation across dependent and independent variables needed for hypothesis testing has been generated using cross-case comparison and meta-analysis techniques, assessing one dependent variable at a time [5, 6].

However, nested conceptual maps and meta-analysis of situation variables will fall short of explaining the functions of instruments in actor networks, The *functions* of instruments need to be analysed in terms of dynamic/adaptive efficiency; whether instruments characteristics help or hinder existing relationships between actors at different governance levels [47][63], calling for qualitative social science methods.

5. Case Studies: Evaluating the Role of Economic Instruments in a Policy Mix

What are the empirical challenges of evaluating the role of particular economic instruments in a policy mix? How can the SES framework be used conceptually and empirically at different spatial scales (landscape level/governmental levels), in addressing different actors (local governments land users) and policy paths? Ecological fiscal transfers (EFT) implemented in Brazil and Portugal provide an example of challenges in assessing effectiveness of economic instruments at different governmental levels. Payments for environmental services (PES) as implemented in Costa Rica exemplify the importance of the policy path and instrument complementarity for the effectiveness of PES.

5.1. Ecological Fiscal Transfers

Public spending includes fiscal transfers between different levels of government – an economic instrument that is still often neglected in conservation policies [64]. Tax revenues are redistributed from national to state and further on to local governments to provide the latter with monies to fulfil their public tasks: building schools and hospitals, or constructing and maintaining roads. Ecological fiscal transfers compensate local and regional governments for the ecological goods and services they provide across local boundaries. Protected areas and local conservation efforts often involve costs at the local level – not least in the form of land-use restrictions – , whereas many of the conservation benefits are provided to higher governmental levels [44, 45]. Ecological fiscal transfers are a suitable instrument for internalising spatial externalities related to the conservation and sustainable use of biodiversity accruing between public actors at different levels of governments.

Only first steps have been made to introduce ecological fiscal transfers by a few countries worldwide: Several states in Brazil have introduced the ecological ICMS, acknowledging biodiversity conservation for the redistribution of value-added tax from state level to municipalities [65-68]. Paraná was the first state in Brazil to introduce the ICMS Ecológico (or ICMS-E, according to May et al.(2002) back in 1992. The ecological ICMS has now been adopted by 12 out of 27 Brazilian states (see Figure 3); others are preparing relevant legislation. The states implemented various ecological indicators for the redistribution of state value-added tax income to municipalities, but Conservation Units (CUs) are the ecological indicator used by all states. Conservation Units are the categories of protected areas for biodiversity conservation in Brazil – their importance as an indicator for redistribution of ICMS-E is weighted by their effectiveness for forest conservation. Portugal has introduced a new national community financing law in 2007 that includes ecological fiscal transfers. The Portuguese fiscal transfer scheme explicitly rewards municipalities for designated Natura 2000 sites and other protected areas within their territories [68, 69].

Figure 3. Twelve states in Brazil have introduced the ecological ICMS that redistributes part of their state value-added tax income back to municipalities based on "Conservation Units".

Both Brazil and Portugal compensate municipalities for land-use restrictions imposed by protected areas and thus combine a regulatory instrument (designated protected areas) with an economic instrument (fiscal transfers). Furthermore, the ecological ICMS in Brazil has developed as an incentive to value and engage in more conservation activities at the local level, including the designation of new protected areas [67] and improving the quality of existing conservation units [70]. For example, in Paraná, since the introduction of the ICMS-E in 1991, Conservation Units increased by 165 % in 2000, and municipalities with larger shares of protected areas considerably benefited from increased fiscal revenues [67].

However, the Brazilian experience also showed that it is important to introduce ecological fiscal transfers with appropriate information and an effective communication strategy. In the state of Minas Gerais, the decentralised organisation of the State Forest Institute helped to publicise the new instrument, along with its task to monitor all information related to ICMS-E transfers based on conservation units [71]. Otherwise local governments and citizens may simply not know that a potentially substantial part of their local public revenues stems from ecological fiscal transfers. Thus, informational instruments clearly contribute to the increased environmental effectiveness of ecological fiscal transfers; although this information does not lead to increased conservation activities by all municipalities that benefit from ecological fiscal transfers [67]. In many cases, this knowledge can reduce local opposition to protected areas and thereby eventually lead to an increase in the quantity of protected areas and, if the indicator for tax redistribution considers the quality of protected areas as it is the case in Paraná, also lead to improving the average quality of protected areas [70]. This also illustrates that economic instruments require a specific information exchange structure whose design will benefit from knowledge of how actors' networks are structured.

In contrast to the rapidly increasing amount of literature on PES schemes, there is relatively little available on evaluating the performance of ecological fiscal transfers. Major empirical research on the ecological ICMS in Brazil has been published by Loureiro (1998, 2002), Bernardes (1999), Grieg-Gran (2000), May et al. (2002), including a more recent review by Ring (2008b). Scientific literature on the Portuguese case is still mostly absent (with an exception of a master thesis by Prates 2008). Further published research on ecological fiscal transfers relates to the design and potential impacts of such instruments that may be newly introduced in contexts where they do not yet exist [72-75].

In this way, ecological fiscal transfers provide an interesting and rather new case for comparative analysis on the effectiveness and efficiency of biodiversity conservation instruments in a policy mix in different SES settings. Whereas in Brazil, economic instruments for compensating the opportunity cost of biodiversity conservation first focused on local governments, other economic instruments directly addressing land users are more recent. In Europe the situation is different. Here, there is long history of compensating land users for their joint production and environmental benefits through agri-environmental payments [68], considered a European variant of PES scheme [76].

In any case, having both EFT for municipalities and PES schemes to land users available poses the important empirical challenge of distinguishing between opportunity cost to local governments and those to the private sector to be able to quantify adequate payments to either municipalities or land users in the local context. Another important empirical challenge relates to quantifying the spillover benefits and the spatial externalities of biodiversity conservation. Although it is widely uncontested that biodiversity conservation involves benefits across local and national administrative boundaries and up to the global level [44, 45, 77], there are few studies that actually try to quantify the spillover benefits of conservation (e.g., an exception is Horton et al., 2003). In this context, benefit transfer studies play an important role, because it will be impossible to evaluate the spillover benefits of biodiversity conservation for any possible setting. The SES framework promises to provide guidance on matches between existing valuation studies and social-ecological context, building on the multiple variables and tiers covered by the framework.

5.2. Payments for Environmental Services

Payments for environmental services (PES) is a category of economic instrument in conservation policy and encompassing a number of variants in country-wide schemes and local projects [7, 20, 78]. Given the wide array of adaptations of the generic PES concept [79], the question arises which contextual factors systematically define the constraints and feasibility of PES. Costa Rica's PES is probably the best documented scheme in the tropical world, providing a good case to discuss the empirical challenges of assessing PES role in a policy mix.

The interaction between PES and other policies on enrolment in PES, effectiveness on land in PES, spillover effects to the surrounding landscape, and the role of PES in 'forest transition' processes have been mentioned by most authors, but few have assessed linkages systematically [80]. Disentangling the effects of Costa Rica's PES programme from that of other conservation policy measures and economic trends is difficult. Aggregate studies of changes in forest cover are difficult to compare as they apply to different areas, time periods, use different dependent variables and methodologies [81].

Costa Rica's 1996 Forestry Law which legally established PES, is itself a policy package of informational, legal and economic instruments: the law defines forest (in terms of percent canopy and area), bans forest conversion to other land uses, restricts logging within 15-50 m (depending on slope) on either side of rivers, and establishes that payments to land-owners for forest conservation and reforestation can be made for carbon, watershed, biodiversity and landscape services (Harvey et al., 2008). Thanks to recent PES targeting, a substantial share of PES contracts are to be found within mixed use conservation units in buffer zones around protected areas and in biological corridors located between protected areas. Alvarado et al. (2009) argue that the success attributed to PES must be interpreted in the context of conservation policies in place, including protected areas, and the general restrictions on timber extraction and forest clearing on private lands. By 2005, 270 000 hectares were enrolled under PES in Costa Rica, 95% of which were for forest conservation activities [81]. In addition, approximately 1 333 000 hectares were in protected areas, with 647 000 hectares in state national parks and biological reserves [82]. To our knowledge the research on the effectiveness of PES has not evaluated the role of location within mixed use conservation units and proximity to strict conservation areas.

Costa Rica's elaborate PES system has been credited with turning one of the world's highest deforestation rates during the 1970s to net reforestation by the early 2000s [81]. Recently, studies have begun to report little significant effectiveness of PES on deforestation, and only small positive results for forest re-growth [80, 83]. Recent research acknowledges that the transition has been built on a series of forest and conservation laws dating back to the early 1970s; and set in the context of economic development that contributed to reducing deforestation pressure [80, 81, 84-86].

The literature evaluating the effectiveness of PES in Costa Rica is extensive, evaluating impact at different scales, including factors affecting farm level adoption [80, 87-89], factors affecting cost-effectiveness of PES targeting at regional level (e.g. [28, 90]), aggregate forest impacts at regional level (e.g. [80, 83, 85]. From the latter three studies a consensus seems to be emerging that PES impact on avoided deforestation/forest conservation has been negligible, whereas its impact on forest regrowth has been small, but significant. Costa Rica's PES system has evolved from its origins as an untargeted and undifferentiated payments scheme that was seen as a quid pro quo for the costs imposed on land owners by legal restrictions on forest conversion [81]. PES targeting has progressively developed to address both conservation (biodiversity corridors) and equity concerns (low socio-economic index areas), with differentiation – admittedly broad – of payments according to types and level of services provided (carbon and hydrological). The modest impact of PES contracts on forest conservation suggests that forest law prohibition of forest conversion, and the concurrent and broad socio-economic development [85] were the 'horse drawing the PES cart'. But it is also a fair argument that the Forest Law in its present form may not have passed without PES as a compensation instrument. The role of PES as a'quid pro quo' also change over time, with recent debate in Costa Rica about the need for PES if deforestation is in fact illegal.

The recognition of a gradual co-development of PES with other conservation policy instruments, and its introduction during a forest transition process, highlights the need to look at socio-economic development context, governance variables and time lags in the study of the cost-effectiveness of PES. Costa Rica's PES was perhaps never a clear policy instrument alternative under consideration by policy-makers, but an evolving package of legal, information and economic instrument characteristics. Complementary approaches to quantitative cost-effectiveness studies are needed to explain the feasibility of different economic instrument characteristics of PES in different settings.

6. Discussion

Based on the recent studies relating the effectiveness of PES and EFT, some conclusions can be drawn regarding approaches to assessing the role of economic instruments in a policy mix.

Cost-effectiveness studies comparing alternative economic instruments versus legal instruments at an aggregate level are complicated by the fact that they have been introduced as part of a policy package from the start (e.g. Costa Rica's 1996 Forestry Law). A cost-effectiveness analysis comparing legal and economic instruments in the Forestry Law would be technically challenged in trying to disentangle *path-dependency* effects[91]. To give a slightly stylised example, the cost-effectiveness analysis would be in danger of ignoring the fact that PES payments were designed as a quid pro quo for the ban on forest clearing, and that the ban on forest clearing across the country was a reaction to weak forest management already in place in e.g. Forest Reserves.

The same holds for Portugal's ecological fiscal transfer scheme that has been introduced in addition to agri-environmental schemes related to EU Agricultural Policy existing for a much longer time. Furthermore, ecological fiscal transfers are designed to build on protected areas, a classic legal instrument of biodiversity conservation, when they use the size of protected areas as an indicator for redistributing tax monies.

The study of the *effectiveness* of economic instruments (EFT and PES) and legal instruments such as protected areas, suffer similar problems of endogeneity in causal factors, for example allocation to high deforestation risk areas. Matching methods have thus far compared deforestation rates in similar spatial units with and without PES, controlling for biophysical, location and socio-economic characteristics [80, 83]. The same techniques should be possible to use in assessing the spatial complementarities between protected areas and PES. However, controlling for the effects of other policy instruments will place great demands on cross-section *sample sizes,* which may not be sufficiently available in any specific region [80]. In most countries with PES schemes in place, time series data on PES implementation is very limited (Costa Rica's 12 years of data represent the most optimistic scenario in this sense). Calvo-Alvarado et al. (2009) demonstrate the importance of evaluating the path dependence of different conservation policies, but the lack of time series data place serious constraints on quantifying these dynamics. Although EFT have first been introduced in Paraná in 1992, due to the rather exotic nature of this instrument in the scientific conservation community, there are fewer data and publications available, and this is even truer for EFT being introduced more recently.

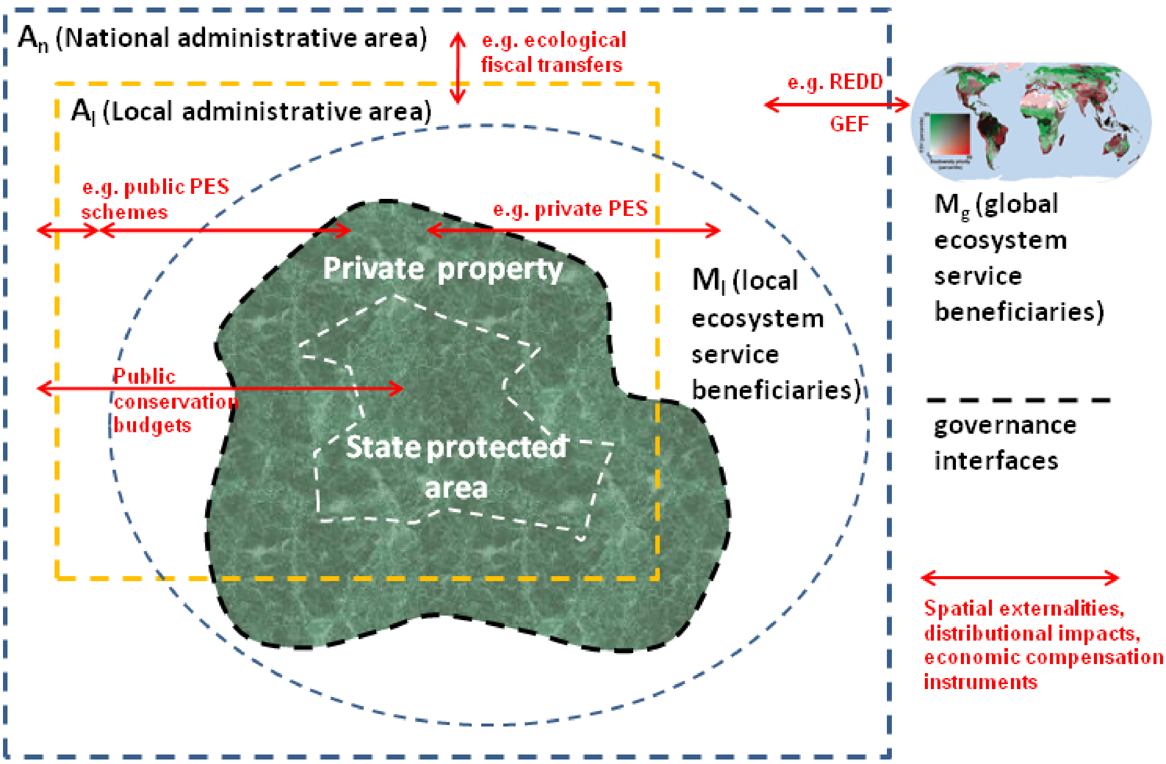
Instruments may also be subject to *other political objectives* than 'narrow' cost-effectiveness. PES in Costa Rica may not have 'additionality' as an objective when they are seen as compensation for obligatory conservation measures such as under the Costa Rican Forestry Law of 1996 [81]. FONAFIFO's approach is to 'recognise' environmental services of whoever is providing them irrespective of the parcel's spatial location, or its capacity to provide a specific and needed service [81]. A proper process of PES negotiation could also be seen as a platform for democratisation and improved governance, motivating the interest of donors [17].

The number of relevant alternative instruments to be considered as part of a policy mix naturally increases with *spatial and temporal scale* of the analysis. When research focus is placed on private farm land, PES constitutes the main conservation instrument in Costa Rica. If other types of private land use are considered, such as tourism, other instruments such as private reserves and conservation easements become relevant alternatives. If geographical and time scale is increased further, different land-use regulations become relevant options for the policy mix; including different categories of state protected areas (Biological Reserves, National Parks) and privately owned multiple use areas (Forest Reserves and Wildlife Refuges, National Wetlands, Hydrological Reserves).).

Fisher et al., (2009) demonstrate how the spatial configuration of landuses in watersheds and landscapes determines the spatial directionality of ecosystem services. We would therefore expect the role and design of multiple use protected areas and PES to be completely different in the absence of "core" protected areas such as National Parks and Biological Reserves. This is why we think an analysis of "roles" is more appropriate than an analysis of "cost-effectiveness" of instruments. The toolbox analogy is once again relevant: (i) it is technically possible to evaluate the cost-effectiveness of a hammer versus a saw in the building of a house, but the analysis would be trivial because the tools were purposively designed to have different roles in different areas of house building; (ii) the cost-effectiveness of alternative designs of a given tool in carrying out its designated role is non-trivial; (iii) there might also be characteristics of different tools which can be recombined to address new requirements of house building. Hence, the ongoing development of PES with new institutions to address the different externality configurations of ecosystem services might be studied as recombinations of *policy package characteristics*, rather than as new instruments per se for example. The SES framework provides a *tiered approach* to such an analysis. A tiered approach would by definition recognise direct and indirect roles of instruments, for example the direct role of environmental education on awareness, and e.g. the indirect role on the willingness to enter into voluntary conservation agreements.

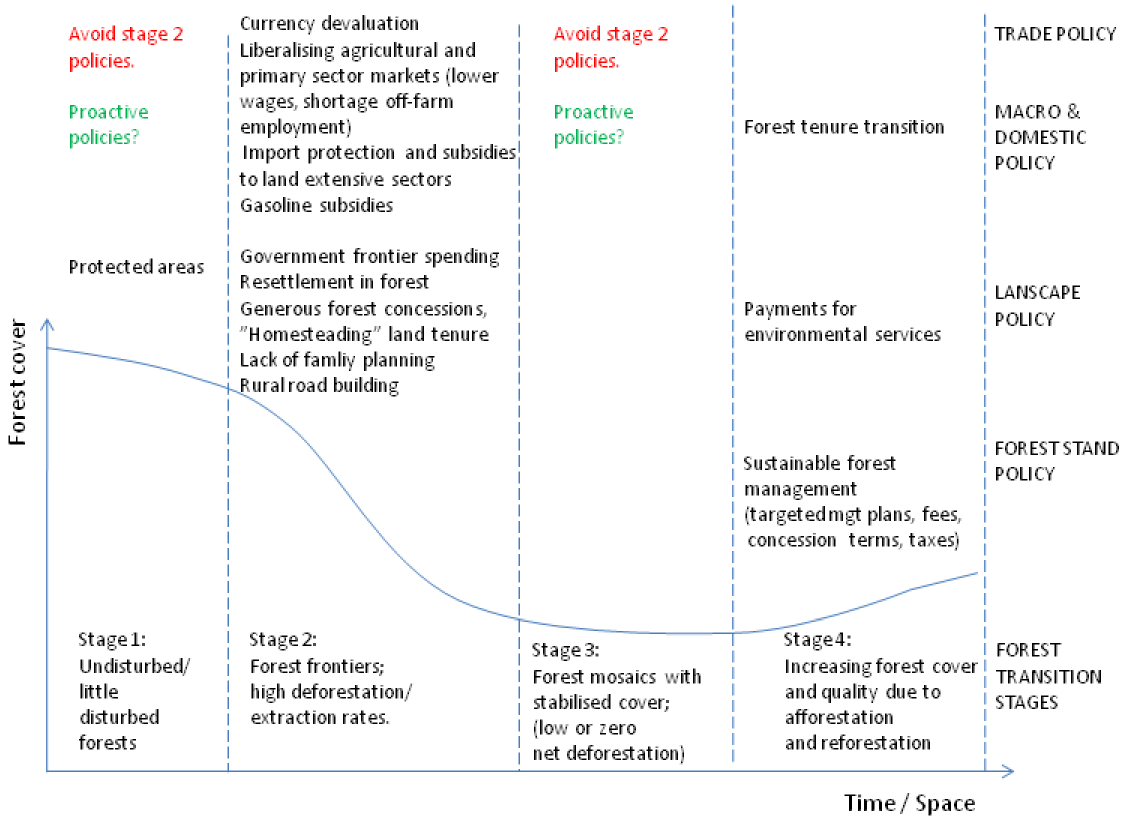
The need for an analysis of the policy mix arises mainly at the *landscape scale* or higher. García-Fernández et al. (2008) assess multiple-use forest management at stand level and landscape level. Their review of a number of land-use modelling studies finds that land-use specialisation at forest stand level, with multiple-use management at landscape level provides superior returns relative to multiple-use management at stand level. Multiple-use at stand level is an option where MFM techniques are low complexity, low cost, typical of extractive uses in hunter-gatherer societies. Richer societies that have passed through a forest transition experienced mixed societal demands on forests that can favour multiple use across the landscape with land-use specialisation at stand level. The set of economic and legal instruments applicable to any particular forest stand, land-use or property rights combination is usually quite restricted, making cost-effectiveness analysis comparing instruments of limited relevance to the owner of a particular forest stand. At landscape scale, overlapping forest characteristics, property rights and administrative levels define a mix of relevant economic instruments. At this landscape level economic instruments play complementary roles by addressing different land-use specialisations and externalities across governance boundaries. Figure 4 shows an idealised example where economic instruments for biodiversity conservation, often seen as alternatives, have complementary roles in the same landscape. The discussion on ‘targeting’ found in the PES literature, focuses mainly on biophysical and farm household characteristics [92]. The SES framework provides a rich backdrop of situation variables which can potentially aid in complementary targeting of economic instruments.

Figure 4. Different economic instruments play different complementary roles across the same landscape, given the configuration of overlapping forest resource, use and governance characteristics. The role of economic valuation is to quantify the externalities (which happen by definition across governance interfaces).



'Forest transition stages' [93] serve as a conceptual framework for understanding the path dependency between the land-use, 'policy- and landscape': (stage 1) multiple forest extractive uses (high forest cover), (stage 2) land use specialization (declining forest cover), (stage 3) specialisation at forest stand level in land-use mosaics (stable low forest cover) and (stage 4) landscape-wide forest management (forest recovery). At the regional and country scale the *forest transitions* concept [94] suggests that appropriate instrument types and combinations depend on the stage along a *development path* [93, 95, 96]. García-Fernández et al. (2008) discuss three stages of transition from multiple forest extractive uses (high forest cover), to land use specialisation (falling and low forest cover), and sustainable forest management (forest recovery) (Figure 5).

Figure 5. Forest transition stages and policy examples at each stage. Research has tended to focus on policies causing changes in forest cover. Part of the policy mix in stage 1 and stage 3 is avoiding policy and institutional failure of stage 2, but are the proactive policies the same as for stage 4? Adapted from Angelsen(2007), Wunder 2003, García-Fernández et al. (2008) and Barbier et al. (2009).



The literature on cost-effectiveness of PES from Costa Rica has begun to report limited effectiveness of PES on forest protection, while positive results for forest growth. Researchers have also begun to recognise that PES might be most adapted for the last stage of forest transition [95]. The literature also largely fails to demonstrate that PES has successfully increased the provisioning of any single ecosystem service in particular. More generally, one can ask whether the economic instruments for biodiversity conservation and the ecosystem services provided by forests can play the same role in forest transition stage for restored forests (stage 4) as with undisturbed forests (stage 1), indeed whether “biodiversity conservation” and “ecosystem services provision” imply different roles for instruments and different policy mixes[97]. Thus far the few studies on impact evaluation of PES has evaluated it merely as a tool for forest protection or restoration, rather than an instrument targeted at particular ecosystem services.

Just as there is no policy panacea for common property resource management [1], there is no single theoretical framework that includes the dimensions needed to evaluate a policy mix for biodiversity conservation and ecosystem service provision. Recently, forest cover change research has emphasised the complementarities of the *temporal* focus forest transition theory with the spatial approach where land rents and distances to markets are determinants of land-use patterns [93, 96]. In extension we think the SES framework and CPRM literature may offer empirical propositions on *underlying factors* driving land rent. Table 1 shows the complementarities between ‘underlying factors’ in land rent models and second tier variables in the SES framework. The land rents approach focuses on explaining agricultural rents relative to forest rents as the dominant driver of forest change [93, 96]. There is clearly a research agenda in combining the literature on PES in agriculture and forestry to address policy mixes that promote stable forest-agriculture mosaics (stage 3) [98].

This should also lead to an increasing recognition that in a *landscape mosaic* there are several processes of forest transition happening in parallel, requiring differentiation and the flexibility to adapt the composition of policy mixes in a particular landscape as it goes through a transition. A hypothesis for cross country comparison, is not only that countries with democratic political institutions are more likely to experience positive forest trends [96], but also that federal and devolved forest governance should fare better than centralised systems. The issue of path dependence shows that both qualitative and quantitative research methods will need to be applied. The SES framework and CPRM literature [1], offers a rich framework from which to draw potential causal variables. A drawback of the SES framework is that it does not indicate a hierarchy of necessary conditions, nor offer causal models. Land rent and forest transition theory does offer such causal models. The literature on pre-conditions for PES springs to a large degree from the land rent approach [7, 17]. It might help define a hierarchy of ‘tiered’ conditions for economic instruments within the SES framework. These hypotheses could also be the subject of further testing in matching studies evaluating the effectiveness of instrument characteristics across the landscape [80]. Given that SES springs from CPRM theory on sustainable common property management institutions, a further research hypothesis is that underlying drivers of land use change on the forest frontier also determine the types of governance systems that appear in stages 3-4 of forest transition. We might speak of a potential framework for evaluating ‘land-use mosaic institutions’[99].

In SES context, culture and education are two aspects interacting/influencing behaviour of individuals. A tiered SES approach will also recognise that instruments have *direct* roles as incentives and *indirect* roles in shaping behaviour through culture and education. For example, agricultural market policies have no direct effects on ecosystem services, but affect the value of alternative land uses to forests; policies promoting education and awareness-raising can in some cases foster voluntarily conservation behaviour.

Beyond exploring in which SES categories the different events and processes that are defined by economic instruments will fall, such analysis should necessarily look for the profile and impact of the *bridge builders* [100]. Questions as what were their roles and how they were able to play them in the context of a new project or instrument are very important to understand, as these kind of inter-institutional interactions are not the rule for state and federal organization environments [101]. Thus, processes, institutions, events and individuals are all parts (at different levels) of the social and ecological system that produced such outcomes.

This discussion also begs the question as to the degree to which policy mix implications of regional land-use gradients may be transferable between cultures and policy contexts, given the importance of *historical moments of opportunity*. PES may stimulate compliance where forest protection would enable proprietors to comply with environmental constraints to which they are legally liable irrespective of initial forest cover (e.g, Brazil's Legal Reserve requirement combined with greater enforcement commitment in the southeast Amazon). Reducing Emissions from Deforestation and forest Degradation (REDD) schemes are said to be most desirable under conditions of high deforestation pressure and high remaining forest cover, yet may be more feasible to implement where deforestation has not reached a serious level (stage 1 in 'forest transition'), but where forest communities are well organised internally and able to marshal REDD payments to stifle exogenous pressures to open up new frontiers. It can be argued that Costa Rica's recent conservation history puts their introduction of PES in 'stage 3' of the forest transition, with protected areas, international terms of trade and economic development having brought the country out of a deforestation path and laid the ground for PES to contribute to forest regrowth ('stage 4'). It was also more politically feasible to implement the FONAFIFO model in Costa Rica in a time at which the nation was actively searching for new sources of foreign exchange, and saw the possibilities inherent in the carbon market with considerable prescience.

7. Conclusions: A Research Agenda

Cost-effectiveness analysis of economic instruments is complicated by the fact that they are usually part of a path-dependent policy package co-developed over time. Legal, informational and economic instruments are often implemented simultaneously and play complementary roles. Other political objectives than cost-effectiveness, such as poverty and fairness concerns may make economic incentives part of sustainable biodiversity governance, although their strict 'additionality' and cost-effectiveness is difficult to demonstrate. Even though economic instruments are targeted at specific land uses there may be spatial spillover effects mutually affecting rates of forest cover change (e.g. between PES and protected areas). Because stand level land-use specialisation is often economically efficient, multiple use forest management, including promotion of environmental services, is more likely in a landscape mosaic. The feasibility of economic instruments targeted at stand level, may be greater in the last stages of forest transition when stand level specialisation within a landscape mosaic is optimal (i.e. PES in Costa Rica). In the first stage before deforestation, landscape level incentives with higher scale, lower resolution may be more appropriate (i.e. such as EFT compensation for protected areas in Brazil).

In evaluating a policy mix a combination of methodologies is needed: the theory of forest transitions to focus on the temporal context of policy, a spatial theory of land rents, and a tiered framework for evaluating governance context and path dependence of sustainable institutions of which economic instruments are a part. We believe that the social-ecological systems may provide a framework for such a synthesis, although this remains to be tested. An approach to decomposing economic instruments into its underlying institutional characteristics would also make it easier to evaluate policy mixes as portfolios of characteristics, rather than instruments per se. This also remains to be tested.

Non-market economic valuation methods are of interest as an awareness raising tool at aggregate level, and perhaps in quantifying spillover effects at larger administrative levels, such as in the context of quantifying ecological fiscal transfers. However, it is hard to see the role of non-market valuation methods in targeting economic instruments within landscapes, or at stand level, given the already considerable methodological challenges in defining (surrogate) indicators for biodiversity and ecosystem services. Defining such surrogate indicators of conservation status is a pre-condition for the relevance of non-market valuation methods in targeting specific types of forest areas for the level of ecosystem services they provide. Until then rough opportunity cost-based approaches will often be sufficient to demonstrate economic feasibility of PES in particular forest stands at farm scales [17].

Cross case comparison of the roles of economic instruments in a policy mix needs a common framework for comparison, which we think SES might provide. To the extent non-market valuation estimates are accepted for policy design (targeting or differentiating service payments), SES variables provide a comprehensive check list for evaluating site similarity and a priori validity of benefits transfer. Latin America’s experience with economic instruments in conservation policy, exemplified by differential implementation of EFT across a number of Brazilian states, and the development of PES in Costa Rica, should provide a number of lessons for Europe, as well.

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50. We use this generic SES framework to organise explanatory variables from several studies evaluating policies to address forest cover change. The exercise is meant to generate discussion on the complementarities between policy analysis approaches, as well as on the causality /hierarchy of explanatory variables for forest cover change, and in turn biodiversity conservation and ecosystem services provision. We return to Table 1 several times in the course of the discussion below.
51. For example, economic analysis of biodiversity and ecosystem services within the context of SES framework requires some further conceptual clarification. Looking at second-tier variables in Table 1, economists would consider "economic value" (RU4) and costs of "human constructed facilities" (RS4) as measures of outcome under efficiency (O1).
52. MA, *Ecosystems and Human Well-Being: Policy Responses. Findings of the Responses Working Group. Millennium Ecosystem Assessment Vol. 3*. 2005, Washington D.C.: Millennium Ecosystem Assessment, Island Press
53. Fisher et al. (2009) write, "The concept of ecosystems services has become an important model for linking the functioning of ecosystems to human welfare. Understanding this link is critical for a wide-range of decision-making contexts. We argue that any attempt at classifying ecosystem services should be based on both the characteristics of the ecosystems of interest and a decision context for which the concept of ecosystem services is being mobilised. We discuss several examples of how classification schemes will be a function of both ecosystem and ecosystem service characteristics and the decision-making context".
54. For example, Engel et al. (2008) adopt a Wunder (2005) definition of Payments for Environmental Services (PES) as (a) a voluntary transaction where (b) a well-defined environmental service (or a land use likely to secure that service) (c) is being 'bought' by a (minimum one) service buyer (d) from a (minimum one) service provider (e) if and only if the service provider secures service provision (conditionality). The definition adequately encompasses the 14 case studies of PES reviewed, but in doing so is too general to provide much

guidance regarding what characteristics are unique to PES and what are common to other proposed economic instruments, or part of the existing mix regulations and economic incentives that affect biodiversity in a particular location.

55. criteria and conditions for policy selection: e.g., static efficiency; dynamic efficiency, complexity, distribution and political issues; asymmetric information and risk, non-point-source pollution; small number of polluters, rent-seeking, developing economy, global pollution; (Sterner, 2003: 214-215); administration and compliance costs (OECD, 1997); political/administrative feasibility (Endres, 1985); fairness and moral considerations as in Field (1994).
56. Design criteria used to characterise CPRMs: "clear boundaries and memberships, congruent rules, collective-choice arenas, monitoring, graduated sanctions, conflict resolution mechanisms, recognised rights to organise, nested units" (Ostrom, 1990: 90).
57. Ostrom, E., *Governing the commons. The evolution of institutions for collective action. political economy of institutions and decisions*. 1990: Cambridge University Press.
58. Endres, A., *Umwelt- und Ressourcenökonomie*. 1985, Darmstadt: Wissenschaftliche Buchgesellschaft.
59. Vatn, A., *Institutions and the Environment*. 2005, Cheltenham: Edward Elgar.
60. Crawford, S.E.S. and E. Ostrom, *The grammar of institutions.* American Political Science Review 1995. **89**(3): p. 582-600.
61. Ostrom (2007) refers to 'situation variables' as the characteristics of the socio-ecological system.
62. Ostrom (2007) writes "Scholars who prefer to collect large samples and use multiple regression or similar statistical techniques are initially horrified when a large set of variables is listed, given the cost of obtaining reliable indicators of the same variable across cultural settings. Mistakenly, they presume that all of these variables need to be measured and included in future research. Instead, third-, fourth-, and fifth-tier variables are potentially relevant only when they are subcomponents of a second-tier variable posited to affect interactions and outcomes."
63. whether the instruments give all power to government or shares it with other actors; whether government reaction is perceived as proportional to a target's actors behaviour; whether instrument is expansive or limitative in terms of the resources given (incentives) to or taken from targets', whether it is voluntary or coercive.
64. Ring, I., *Ecological public functions and fiscal equalisation at the local level in Germany*. Ecological Economics 2002. **42**: p. 415-427.
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